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Supplementary Information

Successful conservation of global waterbird populations depends on effective governance

Amano, T., Székely, T., Sandel, B., Nagy, S., Mundkur, T., Langendoen, T., Blanco, D., Soykan, C.U. and Sutherland, W.J.

Supplementary Discussion

Earlier assessments of global biodiversity changes

In this study we compiled time-series abundance data on 461 waterbird species at 25,769 sites in 132 countries and quantified the global distribution of abundance changes at $1^\circ \times 1^\circ$ resolution, covering areas with varying levels of anthropogenic impacts, socioeconomic status and conservation efforts. This allowed us to (i) identify the hotspots of abundance losses in areas where the biodiversity status has been little known, such as Central/Western Asia, and (ii) detect the effect of spatial drivers on abundance changes, including countries' governance, surface water changes and the effectiveness of protected areas. Although our dataset covers only waterbird species, waterbirds inhabit almost all of the over 40 types of coastal, inland and human-made wetlands, thus representing an ideal indicator taxon for quantifying the status of biodiversity in wetlands, one of the most biodiverse and productive, yet threatened ecosystems^{1,3}.

Earlier attempts to assess global patterns of biodiversity changes are largely categorised into three types. The first type of studies use time-series data to derive general patterns in changes across the globe, such as global-level changes in species abundance¹¹, temporal changes in α diversity and β diversity⁷⁴ and a global temporal trend in marine species richness⁷⁵. Despite the importance of deriving general patterns across the globe, one limitation in this type of studies is that the coarse resolution of assessments makes it impossible to identify the hotspots of biodiversity loss and assess the effect of local-scale drivers, such as habitat changes and conservation efforts.

The second type of studies aims to test the effect of drivers on abundance changes. For example, Vellend et al.⁷⁶ assessed the effect of broad categories of habitat types and drivers on temporal changes in local-scale plant diversity, while Barnes et al.¹³ tested the effect of local-scale drivers and species characteristics on wildlife population trends. These studies, while providing important insights into our understanding of drivers of species abundance

changes, typically suffer from serious gaps in data availability over space, which limits the inferences that can be derived. For example, in Vellend et al.⁷⁶, there was no survey plot in Central/Western Asia and West Africa and only a few in South/Southeast Asia, but these can be the areas where biodiversity is threatened the most (as is also shown by our study). Data used by Barnes et al.¹³ are limited to populations within protected areas, making it impossible to compare trends between protected and unprotected populations. Further, the absence of systematic surveys underlying these data means that the composition of the species monitored and the methodology can vary enormously among sites, and does not allow population trends to be compared over space at the fine resolution our study achieves.

The third type of studies does not use time-series data but instead other information sources to infer biodiversity changes at the global scale. For example, Rodrigues et al.⁷⁷ quantified global trends in vertebrate conservation status, using the Red List Index. The Red List Index reflects changes in species extinction risks based on the Red List assessed by the IUCN, and thus does not directly measure actual changes in species abundance. Meanwhile, Newbold et al.⁷⁸ estimated the loss of species richness and abundance, using non time-series data with the space-for-time substitution method. While this has enabled an unprecedented geographic and taxonomic coverage in the assessment, it is known that the space-for-time substitution method can derive biased estimates of temporal changes⁷⁹.

Also note that while an increasing number of studies have evaluated the effectiveness of protected areas at the global scale^{80,81}, few have measured the effectiveness in terms of actual changes in species abundance.

Population-level effects of explanatory variables

Governance was also the best predictor of population-level changes in abundance, with its coefficients being significant in 55 out of 293 species (18.8%), followed by GDP growth rates (41 species, 14.0%), temperature changes (26 species, 8.9%), precipitation change (21 species, 7.2%), surface water change (20 species, 6.8%) and protected area coverage (two species, 0.7%: Extended Data Fig. 4). Population-level effects of governance, GDP growth rates and surface water change were largely consistent with their species-level effects. Over 60% of the 55 species with significant governance effects showed more population-level declines in areas with less effective governance. Similarly, about 60% of the 41 species showed more declines in areas with rapid economic growth while 70% of the 20 species showed more declines in areas with high water loss. The result that protected area coverage

barely explained population-level changes suggests that local coverage alone may not be sufficient, and extensive coverage at the species level is necessary.

Sensitivity analyses

(i) Correlation between governance and GDP per capita

Although governance and GDP per capita were correlated (Extended Data Table 2), GDP per capita was not associated with community-level changes when incorporated instead of governance (Extended Data Fig. 5a), suggesting that governance, not wealth, explained the global pattern in community-level changes. However, replacing governance with GDP per capita resulted in a significant interaction with protected area coverage at the species level (Extended Data Fig. 5b). Thus the effectiveness of protected areas may also be explained by the wealth of countries at the species level. Nevertheless, this does not change our conclusion that the effectiveness of protected areas is conditional upon better legislation, investments, enforcement and management.

(ii) Designation years of protected areas

Restricting the analyses to protected areas known to have been designated before 1990 (i.e., the first survey year in our dataset) did not change the result: protected areas' effectiveness remained dependent upon governance (Extended Data Fig. 5c, d).

(iii) Inclusion of seabird species

Both the IWC and CBC are not necessarily targeted at seabird species, but our analyses included a total of 14 species in Alcidae, Procellariidae and Sulidae (see Supplementary Data S2 for more detail) because these species are continuously monitored either from land or from boats using constant survey methods, and we also excluded sites only with incidental observations of rare species by only using grid cells with, on average, four or more non-zero records per site. Nevertheless, to assess the effect of including these seabird species in the analyses, we have re-conducted both the driver analyses after excluding the 14 seabird species. Both the hotspots and important predictors of abundance changes remained the same (Extended Data Fig. 6).

(iv) CBC survey sites not targeted at waterbirds

In contrast to the IWC, the CBC is not specifically targeted at waterbirds and its sites in North America may be biased towards urban areas and thus, not necessarily located near wetlands.

However, the proportion of urban areas (calculated based on ⁶¹) in North American grid cells was not particularly higher than that in other regions covered by the IWC (median = 0.005 (North America), 0.008 (South America), 0.015 (Europe), 0.003 (Africa), 0.005 (Western/Central Asia), 0.007 (South/Southeast Asia), 0.035 (East Asia), and 0.004 (Oceania)). Further, we also re-conducted both the driver analyses twice, first after excluding the 41 CBC grid cells which contained neither landscape-scale wetland areas (i.e., were without wetland areas based on ⁶¹) nor local-scale surface water occurrences¹⁶ (within 1km of all the survey sites included), and second, after excluding the eight CBC grid cells which had an especially high proportion of urban areas (the proportion of urban areas over 0.3 based on ⁶¹). Both the hotspots and important predictors of abundance changes remained the same (Extended Data Fig. 7).

Spatial coverage of our data

Although the spatial coverage of our data is not complete, with gaps especially in Mexico, sub-Saharan Africa, New Guinea and the Pacific islands (Extended Data Fig. 1), our results suggest that governance, surface water change and economic growth can potentially predict situations in these data-poor regions. For instance, the Democratic Republic of the Congo, Chad and Nigeria are undergoing rapid economic growth but classified as less effectively governed⁸², while harboring a rich diversity of waterbirds⁸³. These countries, in addition to other hotspots identified in this study, thus merit urgent attention as areas of potential loss and degradation of waterbird populations. Similarly, other data-poor regions merit attention, such as parts of East Asia, where rapid economic growth has been causing drastic changes in inland and coastal wetlands, and as a result, declines in many migratory waterbirds⁸⁴. While the strong effects of some predictors revealed in this study, such as governance and economic growth, indicate that direct and indirect impacts of these factors could override those of other factors in operation at other life history stages of migratory species, incorporating information at different stages of species' annual cycle, although not possible due to the lack of necessary data in this study, would enable us to better explain geographic patterns in declines (e.g., habitat loss at staging sites in East Asia explains declines in migrants in Australia and New Zealand⁸⁵, Extended Data Fig. 2a).

Spatial biases in our data

Although we estimated the distribution of species abundance changes with unprecedented spatial extent (including countries in data-poor regions, such as 21 in Africa, 11 in Central/Western Asia and 14 in South/Southeast Asia), the distribution of our survey sites is

still inevitably biased towards some regions, such as Europe and North America (Extended Data Fig. 1). As the calculation of mean abundance changes at the species or global level could be affected by such spatial biases in survey sites, it is not appropriate to discuss the mean rates of abundance changes at the global or species level based on our data, and thus we avoided making such inferences. Moreover, the objectives of this study are to identify hotspots and drivers of abundance changes, and our main conclusions (i.e., hotspots of declines and importance of governance in explaining abundance changes and protected area effectiveness) should not be affected by the spatial bias in survey sites for the following four reasons.

First, we identified the hotspots of abundance declines based on community-level changes at the grid cell level, not the number of grid cells, thus the number of grid cells available in different regions does not affect the conclusions (e.g., Western/Central Asia has fewer grid cells compared to Europe, but was clearly identified as the hotspot of declines). Of course a complete lack of sites in some areas could still cause other hotspots to be overlooked; see the discussion in *Spatial coverage of our data* above.

Second, in any regression-type analyses, values of each predictor sufficiently covering its plausible range is much more important than using unbiased samples over the range. This applies to all of our driver analyses. For example, even though the number of grid cells used in the community-level analysis is much higher in areas with more effective governance, as long as there are some grid cells in areas with less effective governance, the analysis can detect the effect of governance. In this sense, our study has a strong advantage over earlier studies, as we have abundance change estimates across areas ranging widely in governance levels (see Fig. 3b) as well as protected area coverage (see Extended Data Fig. 3a), making it possible to detect the effects of these two important drivers. Earlier studies were not set up to do this due to the lack of data in, e.g., areas with less effective governance⁷⁷ and outside protected areas¹³.

Third, in the species-level analysis, mean values of each predictor were calculated based on grid cells where each species was actually observed, not the overall geographical range of the species, and thus are correctly matched up with the species-level abundance changes estimated based on those grid cells (see *Driver analysis* in Methods for more detail).

Fourth, we used conditional autoregressive (CAR) models to account for spatial autocorrelation in both the driver analyses. CAR models have been shown to enable precise parameter estimates under the presence of spatial autocorrelation in model residuals⁸⁶. In the framework of this model, an abundance change in a particular grid cell is partly explained by spatial autocorrelation (i.e., abundance changes in surrounding cells), thus adopting CAR

models is effective for alleviating undesired impacts of over-sampling in particular regions on parameterisation.

Potential effects of changes in survey efforts over time

Estimates of abundance changes could be affected by changes in survey efforts in the following two forms: changes in (i) the number of survey sites and (ii) survey efforts within each site across the survey years. Nevertheless, the main conclusions of this study should not be affected by these issues as explained below.

(i) Changes in the number of survey sites

First, mean changes in the number of survey sites over the survey years within each grid cell were largely negligible for most of the 775 populations analysed (median = 0.31, 95% quantile range: -0.33 – 3.00). This shows that the number of survey sites has neither increased nor decreased enormously in most populations and thus species.

Further, more importantly, even for those grid cells with changes in the number of sites, the effect of such changes is already taken into account in the modelling approach. In the model used for quantifying abundance changes, the site effect is included to account for absolute differences in abundance among sites. Therefore, for example, the level of abundance in newly-added sites is modelled separately from other sites, and the change in the number of sites within each grid cell itself should not affect the estimated trend of the cell.

(ii) Changes in survey efforts within each site

First of all, in the IWC, the importance of keeping survey efforts (e.g., the duration and coverage area of each survey) constant is repeatedly stressed and strictly encouraged through, for example, training courses and dissemination based on the materials below.

IWC in general: [https://europe.wetlands.org/wp-](https://europe.wetlands.org/wp-content/uploads/sites/3/2016/08/Protocol_for_waterbird_counting_En_.pdf)

[content/uploads/sites/3/2016/08/Protocol for waterbird counting En .pdf](https://europe.wetlands.org/wp-content/uploads/sites/3/2016/08/Protocol_for_waterbird_counting_En_.pdf)

UK (training courses are also intensively organised: <https://www.bto.org/volunteer-surveys/webs/taking-part/training>): <https://www.bto.org/volunteer-surveys/webs/taking-part/counter-resources>

Ireland:

<http://www.birdwatchireland.ie/OurWork/ResearchSurveys/IrishWetlandBirdSurvey/Methods/tabid/112/Default.aspx>

France: <http://www.oncfs.gouv.fr/Identification-et-comptage-des-oiseaux-deau-en-Afrique-ru535/Identification-et-comptage-des-oiseaux-deau-en-Afrique-ar1425>

Atlantic coast of Africa: <http://www.waddensea->

secretariat.org/sites/default/files/downloads/monitoring_strategy_eaf_3.pdf

South Africa: http://cwac.adu.org.za/instructions_protocol.php

Australia: <http://www.birdlife.org.au/projects/shorebirds-2020/counter-resources>

Indonesia:

<http://archive.wetlands.org/LinkClick.aspx?fileticket=DVw1OYK66N8%3d&tabid=56>

India: http://south-asia.wetlands.org/LinkClick.aspx?fileticket=BI_I_-

i11wc%3d&tabid=2895&portalid=13&mid=12225

Japan: workshops are regularly organised to inform and train surveyors (e.g.,

http://www.bird-research.jp/1_event/shigichi2017_03.html)

Neotropical Waterbird Census: workshops and training courses are regularly organised to inform and train surveyors.

Second, as shown in the method section and Supplementary Methods, we also developed a model to account for temporal changes in survey efforts in the CBC, where the total number of survey hours per count is available, and showed that accounting for changes in survey efforts in the CBC made little difference in spatial patterns in estimated abundance changes. This is presumably because changes in survey efforts in the CBC were moderate; in 84% (137) of the 164 species surveyed in the CBC, over 80% of sites did not show a statistically significant change in survey efforts over the survey periods. This result also suggests that the level of changes in survey efforts that was observed in the CBC would not affect the spatial distribution of estimated abundance changes for waterbirds in general, partly supporting the validity of the model without the effort effect used for the IWC data.

Finally, we cannot dismiss the possibility that the magnitude of changes in survey efforts in the IWC is much greater than that in the CBC and this has affected the abundance changes estimated from the IWC data. However, in order for the effort effect to have falsely created the overall spatial patterns identified in this study, there would have had to be (i) large consistent changes in survey efforts over time at most of the survey sites and (ii) systematic biases over space and species in patterns of survey effort changes. The risk of both of these occurring at the same time in the IWC data is expected to be very low. For example, the spatial pattern of community-level changes (Fig. 2a) consists of a large number of very common species (e.g., mallard, common teal, ruddy shelduck, greylag goose, great white egret and common crane; see Supplementary Data S1 for more detail) showing severe declines in Western/Central Asia and South America while increasing in other regions. It is extremely unlikely that this pattern was falsely created by severe consistent declines in the

rate of detecting these large, easily observable and identifiable species in Western/Central Asia and South America. Similarly, in order for changes in survey efforts to have falsely created the interaction effect of governance and protected areas shown in Fig. 3c, survey efforts would have had to consistently decrease only for species with high protected area coverage in areas with less effective governance. We thus conclude that the risk of the effort effect alone creating all of the important findings in this study is extremely low.

References

- ⁷⁴ Dornelas, M. et al., Assemblage time series reveal biodiversity change but not systematic loss. *Science* **344**, 296-299 (2014).
- ⁷⁵ Elahi, R. et al., Recent trends in local-scale marine biodiversity reflect community structure and human impacts. *Curr. Biol.* **25**, 1938-1943 (2015).
- ⁷⁶ Vellend, M. et al., Global meta-analysis reveals no net change in local-scale plant biodiversity over time. *Proc. Natl. Acad. Sci. U. S. A.* **110**, 19456-19459 (2013).
- ⁷⁷ Rodrigues, A. S. L. et al., Spatially explicit trends in the global conservation status of vertebrates. *PLOS ONE* **9**, e113934 (2014).
- ⁷⁸ Newbold, T. et al., Global effects of land use on local terrestrial biodiversity. *Nature* **520**, 45-50 (2015).
- ⁷⁹ França, F. et al., Do space-for-time assessments underestimate the impacts of logging on tropical biodiversity? An Amazonian case study using dung beetles. *J. Appl. Ecol.* **53**, 1098-1105 (2016).
- ⁸⁰ Gray, C. L. et al., Local biodiversity is higher inside than outside terrestrial protected areas worldwide. *Nat. Commun.* **7**, 12306 (2016).
- ⁸¹ Gill, D. A. et al., Capacity shortfalls hinder the performance of marine protected areas globally. *Nature* **543**, 665-669 (2017).
- ⁸² Amano, T., Country-level relationship between GDP growth rates and governance. *figshare*, <https://doi.org/10.6084/m9.figshare.5417755.v1> (2017).
- ⁸³ Williamson, L. et al., Areas of high diversity for the world's inland-breeding waterbirds. *Biodivers. Conserv.* **22**, 1501-1512 (2013).
- ⁸⁴ MacKinnon, J., Verkuil, Y. I., and Murray, N., *IUCN Situation Analysis on East and Southeast Asian Intertidal Habitats, with Particular Reference to the Yellow Sea (including the Bohai Sea). Occasional Paper of the IUCN Species Survival Commission No. 47.* (IUCN, Gland and Cambridge, 2012).
- ⁸⁵ Studds, C. E. et al., Rapid population decline in migratory shorebirds relying on Yellow Sea tidal mudflats as stopover sites. *Nat. Commun.* **8**, 14895 (2017).
- ⁸⁶ Dormann, C. F. et al., Methods to account for spatial autocorrelation in the analysis of species distributional data: a review. *Ecography* **30**, 609-628 (2007).

Supplementary Notes

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Supplementary Data

Supplementary Data S1 (separate file)

Species-level maps of distribution of population-level changes in abundance for 461 waterbird species analysed in this study. In these maps, dots represent $1^\circ \times 1^\circ$ grid cells with change estimates. In each species, population-level declines that are categorised as Critically Endangered based on the International Union for Conservation of Nature (IUCN)'s Red List criterion A2 ($\geq 80\%$ past decline over three generations) are shown in red, those categorised as Endangered ($\geq 50\%$ past decline over three generations) in dark orange, those categorised as Vulnerable ($\geq 30\%$ past decline over three generations) in pale orange, and other (less severe) declines in yellow. Population-level increases are also categorised using the same ranges as the categories for declines. Changes with 95% credible intervals overlapping with zero are shown with small white dots. Species' non-breeding geographical ranges based on the bird species distribution maps of the world are shaded.

Supplementary Data S2 (separate file)

The list of 461 waterbird species analysed in this study. The explanations and references of column names are as follows: BLsciname: scientific name, BLengname: English name, BLfamily: family name, BLgroup: taxonomic groups defined by the BirdLife International³⁴, IOCgroup: taxonomic groups defined by the International Ornithological Congress³⁶, PHYsciname: scientific name used in phylogenetic trees, PHYlabel: labels used in phylogenetic trees⁵⁵, SISRecID: unique IDs assigned to each taxonomic entity³⁴, Npop: number of regional populations analysed, Nsite: total number of survey sites included in the analysis, Nrecord: total number of count records analysed, Fyear: first observation years, Lyear: last observation years, Ncell: number of $1^\circ \times 1^\circ$ grid cells with count records, BodyMass: body mass (g)⁷², MigStatus: migration status, RangeBrRes: breeding/resident geographical range size (km²), GenLength: generation length (years), IUCNstatus: the Red List category by the International Union for Conservation of Nature (<http://datazone.birdlife.org/home>).